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The Worth of Groundwater Monitoring at a Sanitary Landfill: A Case Study

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ABSTRACT

The aim of the research reported here has been to develop a framework for valuing the information gained from groundwater monitoring at sanitary landfills. The impetus for the analysis was the DEQE proposed regulation requiring three downgradient monitoring wells at all sanitary landfills in the state. This requirement means increasing the number of downgradient wells for most landfills in the state -- most presently have two such wells. A Bayesian decision theoretic approach was developed to assess whether the addition of a third downgradient well was worth more than it cost.

The town of Amherst already has three downgradient wells and was selected as a case study to determine whether its third monitoring well was economic. The results suggest that this third well is, indeed, worth its cost. Sensitivity analyses were performed to determine whether the conclusion remains valid for communities with conditions quite different from Amherst's. These results are tabulated in Section 4 of this bulletin.

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Darleen Slysz prepared this manuscript. No matter how many times we have seen it done -- a cluttered, handwritten manuscript with inserts in margins transformed artistically and seemingly overnight into a manuscript pleasing to the eye -- we continue to be in awe.

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THE WORTH OF GROUNDWATER MONITORING AT A SANITARY LANDFILL: A CASE STUDY

Mary Postle, Cleve Willis, Daniel Dudek, and Loren Lidsky

1. Introduction

Massachusetts relies in no small way upon its groundwater supplies to satisfy growing demands for water. Furthermore, comfort is often taken in the knowledge that groundwater sources exist as backup supplies should they be needed in the future. However, leachate contamination from hazardous or sanitary waste disposal sites serves as a major threat to these vital groundwater aquifers. Within Massachusetts alone, there have been numerous incidents of municipal and private well closures caused by landfill leachate. In most of these cases of groundwater pollution, the first indication of a problem was the finding of contamination in water supply wells.

Contaminated groundwater is exceedingly difficult and expensive to clean up (Jerome, 1983); hence the most expedient approach to groundwater protection is to prevent contaminants from reaching it in the beginning. This need for water quality control criteria for sanitary landfills has been recognized by state and federal authorities. The Department of Environmental Quality Engineering (DEQE), the Massachusetts agency responsible for regulation of sanitary landfills, has drafted a set of proposed regulations intended to prevent or reduce the extent of degradation of groundwater. One section of the proposals requires a minimum number of monitoring wells in operation at a landfill site. These monitoring wells would be used to detect leachate contaminants in groundwater by providing a means of recognizing and determining the size of a leachate plume before it spreads beyond the landfill facility boundary.

Monitoring wells are not required at landfills presently. In practice, though, most have one upgradient (in the direction of decreasing static head) well and two downgradient wells at their perimeters. In the realization of the value monitoring information may have, DEQE has proposed a regulation that would require a minimum of one upgradient and three downgradient wells. For the requirement of an additional downgradient well to be rational from an economic perspective, the expected worth of the additional information on water quality parameters must exceed the costs associated with the additional well.

The natural question is whether the benefits of the additional monitoring exceed the costs. If so, would even more stringent regulations pass the test of economic rationality? To quantify the costs associated with monitoring systems is simple; valuing the benefits of monitoring is much less so. The present purpose is to apply a Bayesian decision theoretic framework to the Amherst, Massachusetts situation in order to estimate the value of the monitoring information available from the third downgradient well; and to determine the theoretical upper limit on the value of a perfect monitoring system.

The remainder of this introductory section is devoted to: describing the leachate contamination problem; outlining the current legislation affecting the regulation of sanitary landfills in Massachusetts;

describing the role of water quality monitoring data; and articulating more formally the objectives of this research.

1.1 *Landfill Leachate.* Landfill leachate is defined as "liquid that has percolated through or drained from landfilled solid waste and which contains soluble, partially soluble or miscible substances removed from waste material" (DEQE, 1983, p. 6). The process by which leachates enter groundwater is slow and dependent on the type of system. Groundwater occurs in two forms -- as confined or unconfined aquifers. An unconfined aquifer is a body of water that occurs in a geologic formation that is fully saturated and has interaction with the water table. An unconfined aquifer is recharged mainly by precipitation, and rises and falls with changes in volume of stored water. A confined or artesian aquifer, however, is enclosed within a consolidated formation of low permeability, such as limestone, and does not have interaction with the water table. The aquifer is completely saturated with water which is under pressure. Recharge to confined aquifers occurs in one or more specific areas, but the water level in the aquifer responds mostly to changes in artesian pressure. Generally, groundwater is recharged by precipitation infiltrating the ground and moving downward until it reaches the saturated zone or water table. Hence, as precipitation falls on a landfill site and infiltrates the soil cover, the liquid that percolates through the solid waste deposits may become leachate.

Once leachate has reached the groundwater system, the movement of the contaminant will be in the same direction as the flow of water. The flow rate of water within an aquifer is highly variable, but typically moves less than two feet per day (EPA, 1977, p. 13). Some of the factors controlling the flow rate of water within an aquifer and its ability to store and transmit water are porosity, permeability, amount of recharge, and hydraulic gradient. As a result of the slow flow characteristic, leachate will move slowly over time in a well-defined plume. The movement of leachate may be vertical or horizontal, depending on the comparative density and natural flow pattern of the groundwater in the aquifer. The rate of movement of leachate is also based on chemical interactions with aquifer materials and changes in water chemistry. Contaminant plumes may deviate from the path of groundwater flow due to differences in contaminant densities. A plume will widen and thicken as it migrates, and some contaminants within the plume may move faster than others. They may also be in liquid phases that are different from that of the groundwater. A plume may be composed of several overlapping layers that vary in the contaminants comprising them.

The characteristics of leachate are primarily a function of the types of waste, amount of infiltrating water, and pH. High concentrations of heavy metals, other inorganics, organics and biological contaminants are common in raw leachate. Table 1 presents concentration ranges for various physical and chemical constituents of typical leachate from municipal solid waste. The constituents that migrate the furthest are those which dissolve readily in water and have a high ion exchange capacity, or which are less dense than the other constituents or the water. These parameters are those that mostly affect the aesthetic qualities of groundwater such as chloride, sodium, and potassium. The heavy metals, complex organics and bacteria, which may have health implications, tend not to migrate very far unless there is a highly permeable soil or free-channel flow as in fractured bedrock. Contaminant

attenuation can occur naturally with time and distance through adsorption, dispersion, diffusion or natural ion exchange. The rate of attenuation or reduction is a function of the types of contaminants and the hydrogeology of the groundwater system. But, it may take decades to remove the contaminants from the water through these natural processes.

Table 1

Leachate Characteristics from Municipal Solid Waste

Components	Median Value (mg/liter)	Ranges of All Values (mg/liter)
Alkalinity	3,050	0-20,850
Biochemical oxygen demand	5,700	81-33,360
Calcium	438	60-7,200
Chemical oxygen demand	8,100	40-89,520
Copper	0.5	0-9.9
Chloride	700	4.7-2,500
Hardness	2,750	0-22,800
Iron, total	94	0-2,820
Lead	0.75	< 0.1-2.0
Magnesium	230	17-15,600
Manganese	0.22	0.06-125
Nitrogen	218	0-1,106
Potassium	371	28-3,770
Sodium	767	0-7,700
Sulfate	47	1-1,558
Total dissolved solids	8,955	584-44,900
Total suspended solids	220	10-26,500
Total phosphate	10.1	0-130
Zinc	3.5	0-370
pH	5.8	3.7-8.5

Environmental Protection Agency, 1977, p. 148.

1.2 *Legislation.* As previously stated, DEQE has proposed a set of regulations for the operation of sanitary landfills in the state (DEQE, 1983). The draft proposals address a number of issues in landfill management such as the siting of landfills, standards for design and operation of facilities, and monitoring for methane gas and leachate contamination. The proposals are, at this point, in the stage of revision after public review. However, the intent of the regulations will not change and one of the main objectives is to protect the quality of groundwater supplies. The regulation of interest to this study would require that all sanitary landfills in the state have a minimum number of monitoring wells at their

perimeters. The requirement is one upgradient well for background quality information and three downgradient wells for detection of quality changes. Currently, no monitoring wells are required by law, but roughly 90% of the landfill sites have wells (Mockrzecky, 1983). Most have at least one upgradient well and two downgradient wells.

The operation of sanitary landfills is regulated on both the state and federal level. The most recent legislation affecting landfills is the Resource Conservation and Recovery Act of 1976 (RCRA). The purpose of RCRA is to provide a set of regulations for dealing with the management of hazardous and solid wastes. The Act charges the Environmental Protection Agency (EPA) with designing a regulatory system for managing both hazardous and non-hazardous wastes. However, the legislative intent concerning non-hazardous wastes was not that they be regulated at the federal level, but that states be assisted in improving their own regulatory efforts. Subchapter IV provided state solid waste agencies with EPA administered grants to use in developing a management plan. The criterion for states to receive a grant was that their plan follow a set of guidelines specified by EPA. The only sanction deriving from non-compliance with the EPA guidelines was the ineligibility of a state to receive the federal planning funds. Although the grant program was discontinued in 1980, the EPA guidelines are still useful to states framing their own management plans.

In the EPA guidelines, section 256.22 (Title 40, Code of Federal Regulations (CFR)) addresses, in part, the question of groundwater monitoring at landfill operations. Subsection (b) states that "monitoring should be required at all facilities, specifically:

(1) Every facility should be evaluated for potential adverse health and environmental effects. Based on this evaluation, instrumentation, sampling, monitoring, and inspection requirements should be established;

(2) Every facility which produces leachate in quantities and concentrations that could contaminate groundwater in an aquifer should be required to monitor to detect and predict contamination."

Under the criteria for solid waste disposal practices, maintenance of groundwater quality is required (40 CFR 257.3-4). The criteria state that "a facility or practice shall not contaminate an underground drinking source beyond the solid waste boundary" or an alternative specified boundary. Maximum contaminant levels are also established for various inorganic and organic chemicals, microbes and radioactive substances.

In an effort to upgrade the present state regulations so as to be consistent with the EPA guidelines and criteria, the DEQE drafted the aforementioned proposed regulatory changes and additions. The current regulations (Chapter 111, Section 150A), enacted in 1971, specify standards for the design and operation of landfill facilities and require DEQE approval of operating plans. A site investigation of the geologic characteristics of a proposed landfill must also be performed. The information in the investigation must include an evaluation of the influence that geologic factors would have on the potential for groundwater and surface water pollution, a determination of the maximum and normal groundwater table elevation and groundwater flow patterns, and

evaluation of the public importance of the groundwater supply to be affected by the operation (310 CMR 19.02:3). Additionally, in the selection of a sanitary landfill site, no area can be considered which does not provide for the protection of all sources of private and public water supplies (310 CMR 19.02:4). However, the current regulations do not adequately address some of the major environmental and management concerns of landfill use. The DEQE proposals update the current law and place greater emphasis on concerns such as groundwater quality maintenance.

For new landfill sites, the proposed criteria are a combination of siting location studies (as now required), installation of suitable landfill liners to control leachates, leachate control systems, monitoring of methane gas generation, and monitoring of groundwater. The requirements regarding leachate control and monitoring are two important aspects of the proposed regulations. Existing landfill facilities must also implement leachate control measures to protect groundwater. However, for these sites control is much harder.

Because many of the existing facilities were designed and constructed with no groundwater protection measures, they are presently generating uncontrolled leachate which in some cases may have already contaminated the water. At some of the sites where water quality is threatened, sophisticated and expensive leachate control technologies will have to be used. Where practical, the installation of liners may be required. However, in most cases some alternate control technology will be needed. Examples of possible actions put forth by DEQE are:

(1) Diversion of surface run-off away from the landfill area to reduce infiltration of precipitation, thereby reducing the generation of leachate;

(2) Diversion and/or collection of surface leachate breakouts via dikes, berms, or interceptor trenches;

(3) Installation of well-points or conduit systems to modify the hydraulic gradient and alter the velocity and direction of leachate migration;

(4) Installation of pumping wells to recover leachate contaminated groundwater;

(5) Installation of low permeability barriers to impede leachate plume migration;

(6) Capping the landfill with a low permeability material to eliminate or reduce the infiltration of rain or snowmelt.

For most sites, the solution will be to cap the landfill with an impermeable material so as to limit the amount of precipitation that can infiltrate the refuse. The more sophisticated technologies will only be required by DEQE when a drinking water source is endangered.

Even with the implementation of leachate control measures, the possibility of groundwater contamination exists. It is for this reason that DEQE proposes to require monitoring of groundwater quality at all facilities. The EPA guidelines allow for contamination within the

facility boundaries, but the DEQE proposals do not. No degradation of any groundwater resource is allowed under the proposed regulations. Because groundwater systems are interconnected, and do not recognize facility boundaries, the DEQE proposal seems more sensible. Also, DEQE maintains that due to the cost and difficulty of implementing a leachate control technology, the prevention of any degradation of groundwater is desirable.

The type, number, and location of wells for a groundwater monitoring system will be determined on a site-specific basis. However, the regulations require that a minimum of four monitoring wells be operated at each facility (DEQE, 1983, p. 30). One of these wells must be located hydraulically upgradient from the landfill perimeter. The well is to be designed so as to yield groundwater samples that are (a) representative of background or upgradient water quality in the zone of saturation near the landfill site, and (b) not affected by the landfill site. Three wells must be located hydraulically downgradient from the landfill perimeter. These wells are to be designed and located so as to intercept any leachate that may be generated by the facility and that may be migrating with the groundwater. Because the four well requirement is a minimum, DEQE would have the option of requiring more wells if the complexity of the hydrogeology of the site demanded it.

The groundwater is to be monitored throughout the operating life of a landfill and for a period of time after closure to be specified by DEQE at time of closure. Samples will be collected from wells and analyzed on a quarterly basis for the first year of the monitoring program. The samples will be used to establish background information on quality parameters. After the first year, the samples are to be collected on at least a semi-annual basis. A minimum set of parameters for sampling and analysis are specified: total iron, chloride, total organic carbon, specific conductance, total organic halogen, and pH. However, DEQE may require other parameters to be sampled or more frequent sampling.

The DEQE proposed requirements with respect to groundwater monitoring are quite similar to the RCRA requirements for groundwater monitoring at hazardous waste landfill facilities. The DEQE regulations, however, are less technically specific than are the RCRA regulations. The monitoring well requirement for hazardous waste landfills is also a minimum of four wells, one upgradient and three downgradient. RCRA goes a step further though and also provides some criteria for the design of the network. The water quality sampling requirements in RCRA are more specific as well, requiring several more parameters to be tested and specifying standards on maximum concentrations of parameters.

1.3 *Role of Monitoring.* Groundwater monitoring is a fundamental part of the effort to protect groundwater quality. Monitoring wells can play an important role in the early recognition of groundwater quality degradation. The main purpose of monitoring is to obtain the type of information from which control decisions can be made. From a pollution management standpoint, the purpose is to obtain an early warning of groundwater contamination and/or to determine the progress of contamination or a change in its components. The information acquired from monitoring wells may mean the difference between taking a remedial action to inhibit further movement of a leachate plume or having to find an alternative drinking supply source for a community.

Monitoring has been defined as the activity involving "systematic observations of parameters related to a specific problem designed to provide information on the characteristics of the problem and their changes with time" (Sherwani and Moreau, 1975, p. 3). The worth of information from water quality monitoring is positively related to its precision or accuracy in predicting water quality parameters and changes. Increasing precision of a monitoring system is accompanied by higher costs, however, and hence the economically optimal monitoring network is not necessarily the one that provides the greatest accuracy or quantity of information. Rather, it is that system which is associated with the greatest benefits of information less costs.

The value or worth of data is defined as a function of its use, and can be estimated by examination of the decisions likely to be made with and without additional sample data, as well as the array of possible consequences of these decisions. As Dawdy *et al.* (1970) and Moss (1970) expressed it, the worth of additional information can be related to the expected benefits foregone as a result of not having and using this additional set of data.

Most monitoring systems have been set up, at best, in a cost-effectiveness manner. That is, given a budget, design a system to maximize the precision of the information; or, given a desired level of precision, minimize cost (budget). Since the precision level (or budget) is established arbitrarily, an optimal monitoring system may differ because the benefits of additional information are considered in selecting a desired precision. In a setting in which information is treated as an economic good, information should be gathered up to the point at which the marginal cost of acquiring it is equal to its marginal worth.

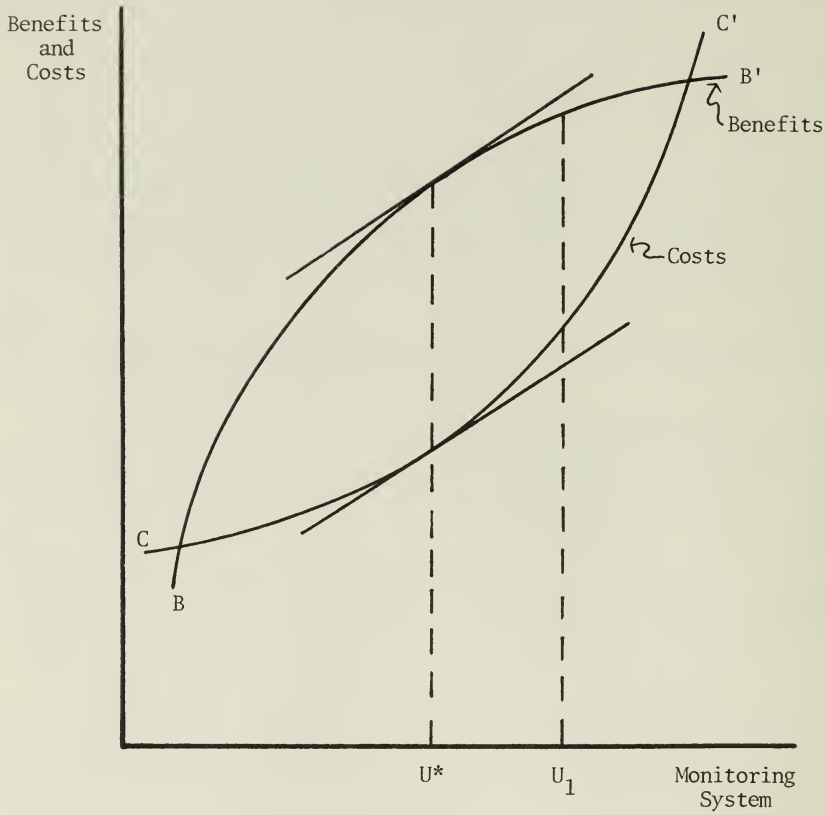
Figure 1 gives an example of the difference between a network which has been set up at the optimal level of monitoring precision and another set at an arbitrary precision level. The BB' curve contains the set of points of maximum value of information given the varying levels of precision. As the precision of monitoring increases, or more data are collected, the marginal worth (slope of BB') of data decreases. The CC' curve contains the set of points representing the costs associated with efficiently gathering the information at different precision levels. The slope of CC' increases at an increasing rate since the marginal cost of acquiring data increases with greater accuracy levels.

If the value of data is greater than its costs of collection, then that specific level of monitoring is economically justified. However, the optimal precision level is given by finding the point where the positive difference between the two curves is the greatest. This corresponds to U* in Figure 1. At this point the slopes of the curves are the same -- i.e., marginal value equals marginal cost.

The optimal level U* can be compared to another level set arbitrarily at U_1. At U_1 the system is still economically justified, but marginal cost is greater than marginal benefits. In other words, if a one-unit decrease in precision occurred from the level U_1, the reduction in costs of monitoring would be greater than the reduction in value of monitoring at that precision level. Hence, it is more efficient to cutback on the precision required and operate at U*.

Figure 1

Optimal Monitoring Network



1.4 *Objectives*. The main purpose of this research is to develop a rationale based on economic efficiency for determining the optimal level of groundwater monitoring precision at sanitary landfills. The specific objectives are twofold:

(1) DEQE has proposed a regulation which would require sanitary landfill facilities in the state to have a groundwater monitoring network consisting of one upgradient well and three downgradient wells. Since most of the small municipal landfills presently operating in the state have two downgradient wells, DEQE has implicitly placed a value on the information gained from the third downgradient well. The first objective is to estimate the expected value of information, using a Bayesian theoretic approach, of this additional well.

(2) After estimating the expected value of information from the additional monitoring well, an economic limit can be found for the monitoring precision level at a landfill. By finding the point at which the expected value of information gained from the monitoring system is equal to the cost of the system, the economically optimal level of monitoring can be determined. The second objective is to establish a limit on the precision level from an economic efficiency perspective.

Section 2 describes the methodology used, Bayesian decision analysis, while Section 3 contains an empirical application using Amherst, Massachusetts as a case study. The results of the empirical application are presented in Section 4, while a discussion of the results and the conclusions from the research are presented in a final part.

2. Bayesian Decision Framework

Bayesian analysis provides a methodology for aiding decision makers in choosing between alternative actions to be taken when faced with uncertainty of occurrence of a set of parameters or states of nature. Within this framework, prior information is combined with additional or sampling information on the states of nature to reduce uncertainty on the occurrence of those states. An example of additional information may be sample information on water quality characteristics acquired from a monitoring well system. By adding the sampling information to what is already believed about water quality, a decision maker can increase the probability of choosing that action which will maximize expected value for a given state of nature. Once the Bayesian analysis has been performed, and the optimal actions given all available information of the states of nature have been determined, the expected value of the information from a given precision level can be found. The worth or value of information derives from reductions in the costs of wrong decisions due to lowered uncertainty.

2.1 *Bayesian Decision Theory.* The theoretical framework for Bayesian analysis is discussed in Chernoff and Moses (1959), Halter and Dean (1971), Raiffa and Schlaifer (1961), and Rausser, Johnson and Willis (1983). Several research applications have empirically determined the value of additional information. In the water resources area, Moss and Karlinger (1974) used Bayesian theory to estimate the benefits from adding an extra unit of time or area covered to a surface water monitoring network. Gates and Kisiel (1974) determined the worth of groundwater data as measured in improved precision of estimated parameters. Maddock (1973) used the framework to find the worth of data through its impact on decisions concerning the management of a groundwater system. Other applications include those of Baquet *et al.* (1976) who derived the economic value of frost forecast information, and Morzuch and Willis (1982) who determined the value of weather information to cranberry marketing firms. Additional examples of agricultural research using Bayesian analysis are given by Anderson *et al.* (1971), Carlson (1970), and Eidman *et al.* (1967).

The basic framework of the theory is as follows. A decision maker must choose an action (a_j) from a set of actions where the consequences (c_{ij}) of his choice depend upon the prevailing state of nature (S_i). The state of nature is unknown, but has a probability of occurrence ($P(S_i)$). Hence, the expected value of an action is the probability weighted sum of the consequences of that action:

$$V_j = \sum_i c_{ij} \, P(S_i). \tag{1}$$

The decision maker then chooses that action with the highest expected value:

$$V = \underset{j}{\text{maximum}} \; V_j. \tag{2}$$

The decision maker begins with only a prior assessment on the probabilities of the states of nature ($P(S_i)$) to help in his choice of action. However, if additional data can be acquired, the decision maker

can reduce the uncertainty about the actual state of nature occurring. The additional or experimental information can be expressed in the form of conditional probabilities ($P(Z_k|S_i)$). These are the probabilities of observing a particular experimental result (Z_k) given that a state (S_i) occurs. By using Bayes' rule, this information is revised to obtain a set of posterior distributions ($P(S_i|Z_i)$) which predict the occurrence of a particular state of nature given that a sample result occurs. Bayes' Theorem is as follows:

$$P(S_i|Z_k) = \frac{P(S_i)\ P(Z_k|S_i)}{P(Z_k)}, \tag{3}$$

where:

$$P(Z_k) = \sum_i P(S_i)\ P(Z_k|S_i). \tag{4}$$

The expected value of each action has now changed because the consequences are weighted with the new posterior probabilities. The expected value of action j given the posterior probabilities is now:

$$V_{jk} = \sum_i c_{ij}\ P(S_i|Z_k). \tag{5}$$

The decision maker will once again choose that action which maximizes expected value. However, this time her choice is based on both the prior information and the experimental results.

The value of the experimental or additional information can then be found by determining the maximum expected value of an action for each k (6) and multiplying each by its probability of occurrence (7).

$$V_k = \underset{j}{\text{maximum}}\ V_{jk} \tag{6}$$

$$V^* = \sum_i V_k\ P(Z_k). \tag{7}$$

Equation (7) represents the expected value of the Bayesian strategy; or, in other words, maximum expected value resulting from both the prior and the sample information. The value of the sample information (8) is the difference between the value of the Bayesian strategy and the maximum expected value resulting from having only the prior information.

$$W = V^* - V. \tag{8}$$

If the value of information exceeds the costs of obtaining it, then it is economically justified.

2.2 *Numerical Example.* An illustration using a simplified numerical example is provided to show how the value of information of a groundwater monitoring network at a sanitary landfill could be estimated. To reiterate the components of the analysis, they are the states of nature (S_i), probabilities of the alternative states ($P(S_i)$), actions (a_j) and

their associated consequences (c_{ij}) which are jointly dependent on state i, and predictors (Z_k).

Assume for this example that water quality can be classified into three levels, where the states of nature (S_i) represent levels such that:

S_1 -- denotes good water quality
S_2 -- denotes acceptable water quality
S_3 -- denotes poor water quality

The actions are the decisions that are made concerning the management of the landfill facility. Assume that three possible actions (a_j) exist for the decision maker to choose. For this example the actions may be represented as:

a_1 -- take no action concerning water quality, operate the facility as usual;
a_2 -- implement some treatment process or leachate control technology;
a_3 -- close affected supply wells and find an alternative drinking source.

The consequences (c_{ij}) of any choice of action depend upon the state of water quality (S_i) that occurs. The consequences are the costs related to the action/state pair. The costs may range from being zero for no action taken to several hundred thousand dollars for finding an alternative water source. The true state of nature is unknown but has a probability of occurrence given by $P(S_i)$. These probabilities come from some prior knowledge the decision maker has concerning S_i. Table 2 depicts the matrix of consequences for every action/state combination and the prior probability of each state.

Table 2

Consequences Based on Prior Information

States	Actions			
	a_1	a_2	a_3	$P(S_i)$
S_1	10	15	20	0.2
S_2	20	15	18	0.6
S_3	25	15	6	0.2
V_j	19	15	16	

The expected value of each action (V_j) can, in turn, be calculated by weighting the consequences of the action given the alternative states of nature by the respective prior probabilities of these states (Morzuch and Willis, 1982, p. 501). Mathematically, the expected value of each action is given by V_j from (1). For example, the expected value of a_1 is:

$$V_1 = 10(.2) + 20(.6) + 25(.2) = 19.$$

In this example, the decision maker will select that action which minimizes the expected costs, V_j. Here a_2 has an expected value of 15 and will be the decision maker's choice. If perfect information (V^*) on which water quality state was occurring existed, the expected value of the decisions would be equal to the summation of the least cost consequence (c_{ij}^*) for each quality level S_i multiplied by its probability $P(S_i)$:

$$V^* = 10(.2) + 15(.6) + 6(.2) = 12.2.$$

The expected value of perfect information (EVPI) is then:

$$EVPI = 15 - 12.2 = 2.8.$$

Now assume that sampling information which is less than perfect is available from a monitoring network at a cost to the decision maker. However, the information can help reduce the uncertainty about the water quality level. With a series of samples at several locations, the true state of water quality can be predicted with some accuracy. For this example, these predictors (Z_k) are classified as:

Z_1 -- no change in water quality;
Z_2 -- small change in water quality;
Z_3 -- large change in water quality.

The sampling information may be expressed in the form of conditional probabilities. These are the probabilities of observing Z_k, a sample result, given that S_i occurs. The conditional probabilities $P(Z_k|S_i)$ of a particular monitoring system will vary with the level of precision. The probabilities associated with a two downgradient well monitoring system will represent a lower precision level than for a three downgradient well system. For this example, a precision level of U_1 is assumed. The conditional probabilities for this system are shown in Table 3.

Table 3

Conditional Probability Matrix

States	Sample Values		
	Z_1	Z_2	Z_3
S_1	0.7	0.2	0.1
S_2	0.3	0.4	0.3
S_3	0.2	0.3	0.5

As long as the conditional distributions are known, they can be blended with the prior probability distributions to obtain a set of joint probabilities. These posterior distributions $P(S_i|Z_k)$ were set

out in (3) above. To illustrate, consider the likelihood that water quality level is poor (S_3) and that sample results indicate that it is good (Z_k):

$$P(Z_1) = (.2)(.7) + (.3)(.6) + (.2)(.2) = .36$$

and

$$P(S_3|Z_1) = (.2)(.2)/.36 = 0.11.$$

The nine posterior probabilities are provided in Table 4.

Table 4

Posterior Probability Matrix

States	Sample Values		
	Z_1	Z_2	Z_3
S_1	0.39	0.12	0.07
S_2	0.50	0.71	0.60
S_3	0.11	0.17	0.33

The original consequence matrix is now augmented using these posterior probabilities to form a posterior payoff table (Table 5). The expected value of each action given these posterior probabilities is given by (5) above. For example, the expected value of action a_3 given Z_1 is computed as:

$$V_{31} = 20(.39) + 18(.5) + 6(.11) = 17.46.$$

Table 5

Consequences Based on Prior and Sample Information

States	Actions		
	a_1	a_2	a_3
S_1	10	15	20
S_2	20	15	18
S_3	25	15	6
V_{j1}	16.65	15	17.46
V_{j2}	19.65	15	16.20
V_{j3}	20.95	15	14.18

From this table, the decision maker can choose which action to pick when a particular sample result is observed. If Z_1 or Z_2 is observed, a_2 will minimize expected costs; if Z_3 is observed, a_3 should be selected.

The expected value of the actions with the sample information can be found by taking the smallest expected consequence value for each k and multiplying them by their probability of occurrence $P(Z_k)$. The sum of these products is the expected value of the Bayesian strategy.

$$V^* = 15(.36) + 15(.34) + 14.18(.3) = 14.75.$$

To determine the value of the sample information from water quality monitoring, the expected value of the Bayesian decisions found using the added data (V^*) is subtracted from the minimum expected consequences of the decision resulting from having only the prior information (V). Hence, the value of the additional information is:

$$W = 15 - 14.75 = 0.25.$$

This gives the maximum monitoring cost which should be incurred on the basis of the reduced costs of consequences from a monitoring system with a level of precision U_1. This monitoring network can be compared with another having a precision level of U_2 by finding the difference between the two expected values of the Bayesian strategy.

3. Empirical Application

The main objective of this research is to determine the value of information generated by sampling from an additional downgradient well as required by the proposed DEQE sanitary landfill regulations. As previously indicated, this value is expected to vary between sites. A case study approach focusing on the town of Amherst was selected for the empirical application in part because it presently fulfills most of DEQE's proposed monitoring requirements and is considered representative of a large number of Massachusetts situations.

In order to perform the empirical application, the states of nature, actions, consequences and probabilities appropriate to the Amherst facility had to be defined. Any classification of these components must take into consideration the characteristics of the landfill site being analyzed. The geology, hydrology, size, and type of wastes deposited are some of the site-specific factors that must be included in the determination and choice of the set of states, actions and consequences. The probabilities will depend on the complexity of the hydrogeology and the monitoring network at a facility. Since the characteristics describing landfill facilities vary, the appropriate states of nature, actions, consequences and probabilities will also vary in the application of the theory to different facilities. Before defining the values used in the empirical analysis, a description of hydrogeology and operation of the Amherst landfill is presented.

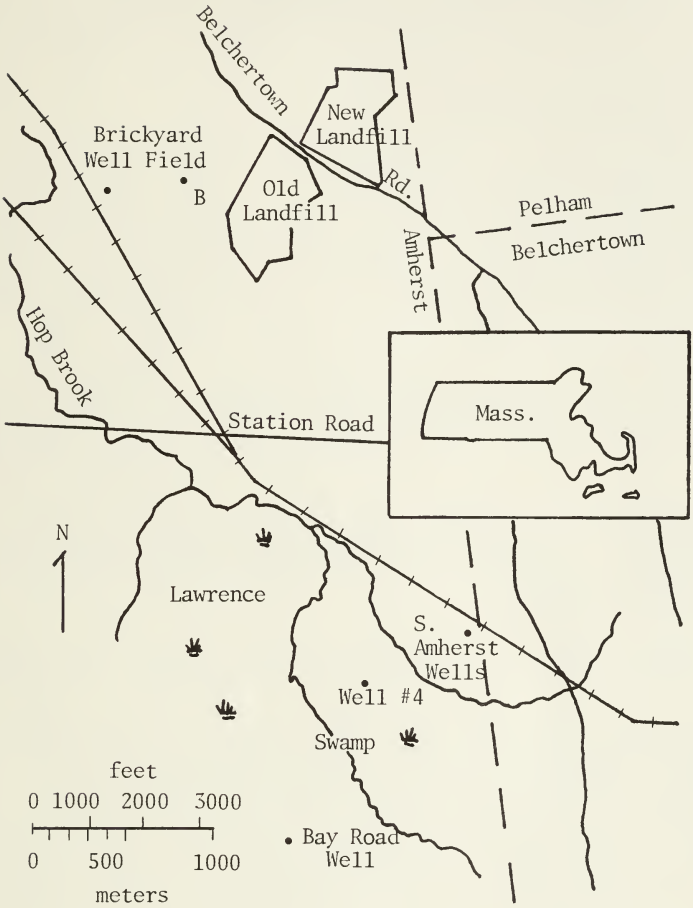
3.1 *Amherst Landfill.* The Amherst sanitary landfill is located on the northern side of Route 9, near the Amherst-Belchertown-Pelham town lines. Municipal wastes from all three towns are deposited at the facility. More specifically, it is situated within the northern segment of the 15.9 square mile Lawrence Swamp Basin. Figure 2 shows the location of the new landfill with respect to the three town lines and the major supply wells in the basin. The supply wells located in Lawrence Swamp, the South Amherst wells, Well #4, and the Bay Road well together supply 65% of Amherst's drinking water demands. Also shown in Figure 2 is the older closed landfill site which was not considered in this study. A leachate contamination problem already exists from the old landfill and has affected the downgradient Brickyard wells, causing them to be closed. As a result of the contamination, intensive monitoring and studying of the site has occurred.

3.1.1 *Hydrogeology of Lawrence Swamp.* The Lawrence Swamp Basin is both a hydrologic and a groundwater basin. The basin is enclosed by topographic divides, so that any precipitation falling within the area either becomes surface runoff or part of the underlying groundwater system. It is a groundwater basin because all of the groundwater in the basin is confined by groundwater divides. Groundwater flows within the confines of these divides discharging into Hop Brook, the master stream of the basin. The groundwater system consists of a confined (artesian) aquifer and an unconfined (shallow) aquifer. The artesian aquifer serves as the water supply source for Amherst's wells.

The principal hydrogeologic units of the basin are: (1) bedrock; (2) till; (3) stratified drift (coarse phase) including the water-producing sand and gravel of the artesian aquifer; (4) stratified drift (fine phase) including clay and fine silt which forms groundwater confining

Figure 2*

Case Study Landfill Location



*Schwalbaum, 1983, p. 3.

units; (5) post-lake beds which are predominantly alluvial sand and gravel and form the shallow unconfined aquifer (Motts, 1982, p. 90).

Bedrock underlies the entire basin. Glacial till mantles the bedrock throughout most of the basin and is considered to be part of the bedrock. The till, which is generally less than 10 feet thick, is composed of non-stratified and poorly sorted deposits consisting of gravel, sand, silt and clay. Although the till serves as a private water source throughout the area, the available water quantity is insufficient for municipal use. However, the permeable upper till (less compact and better sorted) absorbs considerable amounts of precipitation and serves as a recharge area where exposed.

The stratified drift, which covers 31% of the basin, forms the water-holding sediments of the artesian aquifer. The lower zone of well-sorted sands and gravel has a high water-yielding capacity. Occurring immediately above the bedrock and till, this zone is found throughout the basin. Beds consisting mostly of fine through coarse sands that are poorly sorted and contain considerable amounts of silt are located above the lower zone. Layers of clay are spread through the sand but are not continuous. Thus they retard, although do not prevent, vertical movement of water.

However, a clay zone occurring above the artesian aquifer acts as an effective barrier to vertical groundwater movement. The clay lens varies in thickness, being thinner at the swamp margins and increasing in thickness towards the center. In the central part of the swamp, very small amounts of water move through the clay due to its low permeability. At the swamp margin, where the clay lens is thin, a greater amount of water can move up or down. This zone serves as a protective barrier for the artesian aquifer. The clay retards downward movement of water from the shallow aquifer, preventing any water that may be contaminated from entering the confined system.

The sediments composing the shallow aquifer are diverse in origin, but are characterized mostly by alluvial sand and gravel. The thickness of these sediments ranges up to 40 feet, with the greater thickness in the central part of the swamp. Only small amounts of water can be held in the relatively thin beds of the shallow aquifer. The Brickyard wells, in contrast, are located in an area with high transmissivity and, before the leachate contamination problem, were pumped from this aquifer. The shallow aquifer is recharged from direct precipitation and by the artesian aquifer. Because the potentiometric head of the artesian aquifer is higher in elevation than the potentiometric head of the unconfined aquifer, water moves upward through and around the confining middle unit (clay zone) into the shallow aquifer. Even though the head of the water in the artesian aquifer lies above the unconfined aquifer and the swamp, it must be remembered that the actual occurrence of the water is in the sand and gravel below the confining clay unit.

The primary recharge areas for the basin are those underlaid by permeable sand and gravel where infiltration capability is high. Less permeable till and bedrock form the areas of secondary recharge. Water moves from recharge areas along the margin of the basin downward into both artesian and shallow aquifers. All water in the artesian aquifer eventually discharges by well withdrawals or into the shallow aquifer and subsequently from the shallow aquifer into Hop Brook and its tributaries. Water leaves the basin by way of the Fort River (Motts, 1982,

p. 12). Precipitation falling on the secondary recharge areas has two fates; infiltrating into and moving through the bedrock eventually discharging into the artesian aquifer, or becoming surface runoff and infiltrating into the primary recharge area or into the shallow aquifer. Precipitation falling on the primary recharge area follows the same two routes. About one-half of the precipitation filters through the highly permeable sands and gravel and enters the groundwater zone, moving into either the confined or unconfined aquifer. Or the precipitation becomes runoff which filters into the unconfined aquifer or leaves the basin via Hop Brook.

As previously stated, the two aquifers are hydraulically connected, the confined aquifer recharging the unconfined aquifer in the central part of the swamp and the unconfined recharging the confined along some of the swamp margins. However, due to leachate contamination from the old landfill in the northern part of the basin, questions about the ability of contaminated water to migrate from the unconfined aquifer toward the supply sources have arisen. The degree of connection between the two aquifers in the area downgradient of the landfill has not been firmly established (Schwalbaum, 1983, p. 114). Yet, it is believed that a massive silt unit, acting as a flow barrier, extends across the region between the contaminated water and the southern area from which the production wells are drawing. The possibility that the silt unit is not continuous and that leachate may migrate into the confined system, however, demands that any further leachate contamination be prevented. This is also important to the new landfill because the potential exists for leachate to enter the groundwater system.

The landfill facility is located in a recharge area for the basin (Motts, 1982, Figure 21). The principal rock types underlying the facility are bedrock and unconsolidated sediments of glacial origin (Metcalf and Eddy, 1980, p. 11). The bedrock is made up of crystalline rocks and has a layer of glacial till above it. The bedrock surface is as little as 6 feet below ground level in the eastern portion of the area and becomes increasingly deeper in a westerly direction reaching a maximum depth of at least 52 feet below the surface.

The glacial sediments mainly consist of stratified drift. These sediments also increase in thickness in a westerly direction, ranging from less than 6 feet to more than 50 feet. The sediments are composed mostly of stratified sand with subordinate layers of gravel. The sands have a greater permeability near the surface where they are coarser, with the permeability decreasing as the sands become finer with depth. The saturated zone occurs in sands primarily of fine to medium texture. The estimated average permeability of the sands is 150 gallons per day per square foot. The unsaturated sandy deposits are expected to permit relatively free movement of percolates to the water table. Yet, due to the sandy makeup, little chemical attenuation of leachates through ion exchange or adsorption is anticipated.

The groundwater body occurs in the stratified drift and is part of the unconfined aquifer system. The water table slopes in a westerly direction from an altitude of about 290 feet near Harkness Road (eastern edge of landfill site) to about 240 feet near the southwest corner of the site. The groundwater flows mainly in a westerly direction at a rate of about 2 feet per day. Movement of the water is directed, in part, toward Harkness Brook, with the remainder believed to discharge into

streams serving as tributaries to Hop Brook and the Fort River. The levels of various water quality parameters found in groundwater of the area are compared to the EPA Interim Primary Drinking Water Regulations limits in Table 6 (Metcalf and Eddy, 1982, p. 3-3).

Table 6

Water Quality Ranges

	EPA Limits	Background
BOD		< 1-25
COD		< 1-160
Alkalinity		10-56
Hardness		16-66
Iron	0.3	0.00-11.8
Manganese	0.05	0.00-1.5
Free Ammonia		0.00-2.9
Nitrate	10	0.00-2.4
Chloride	250	2-30
Sulfate	250	5-20
Calcium		5.8-12
Magnesium		1.5-2.8
Sodium		2.5-18
Potassium		0.5-2.2
Copper	1.0	0.00-0.12
Lead	0.05	< 0.05-0.01
Zinc	5	0.19-3.0
Cadmium	0.01	0.00-0.01
Fluoride		< 0.1
Silver	0.05	< 0.02
Total Dissolved Solids	500	48-160
Specific Conductance		55-200

3.1.2 *Amherst Landfill Operations.* At the present time the landfill is in Phase I of its development. The total development plan calls for three phases totaling 32 acres in area, with each phase corresponding to a section for fill. Phase I is a 7-acre section located in the southeastern portion of the landfill facility. The section is lined with a 12-inch thick clay liner covered with dense pea gravel. The liner is graded to a saucer-like shape so that leachate collecting on the surface will migrate towards drainage or collection pipes in the liner. The base of the landfill surface must be at a minimum of five feet above the highest level of the underlying water table.

The Phase I section is broken up into individual cells for filling at one time. The cells are 40 feet wide and have no lining material between them. Within a cell, refuse is spread and compacted in layers two feet deep, with a daily maximum of nine feet. The section is covered by 9 feet of fill throughout before starting again on the next layer. Six inches of compacted material must be spread over the refuse at the end of each day. The final cover for the refuse must be 24 inches of material, with at least 12 inches being impervious. As the surface of the

refuse rises, berms are to be constructed so as to prevent any precipitation from ponding on the refuse surface and soaking in.

3.1.3 *Groundwater Monitoring*. Four monitoring wells exist for water quality sampling. One is the upgradient, background information well. This well is located on the eastern side of the Phase I section. The three downgradient wells are located in a triangular pattern, bordering the southwestern corner of the site. All of the downgradient wells monitor at approximately 30 feet into the underlying system, although their surface elevations vary. The locations of the wells were determined by DEQE.

Sampling from the monitoring wells takes place once every six months. Tighe and Bond, an engineering consulting firm, is hired by the town of Amherst to perform the tests. The test results are sent to both the town and DEQE. The sample parameters have been specified by DEQE and are as follows:

Heavy Metals	Sanitary Constituents
Arsenic	Chloride
Barium	Specific Conductance
Cadmium	Manganese
Mercury	Alkalinity
Chromium	Hardness
Nitrate	Iron
Selenium	pH
Silver	
Lead	

At this point, it is interesting to note that while several of the parameters required by Amherst are additions to the required minimum set proposed by DEQE, testing is not required for two parameters: total carbon and halogens.

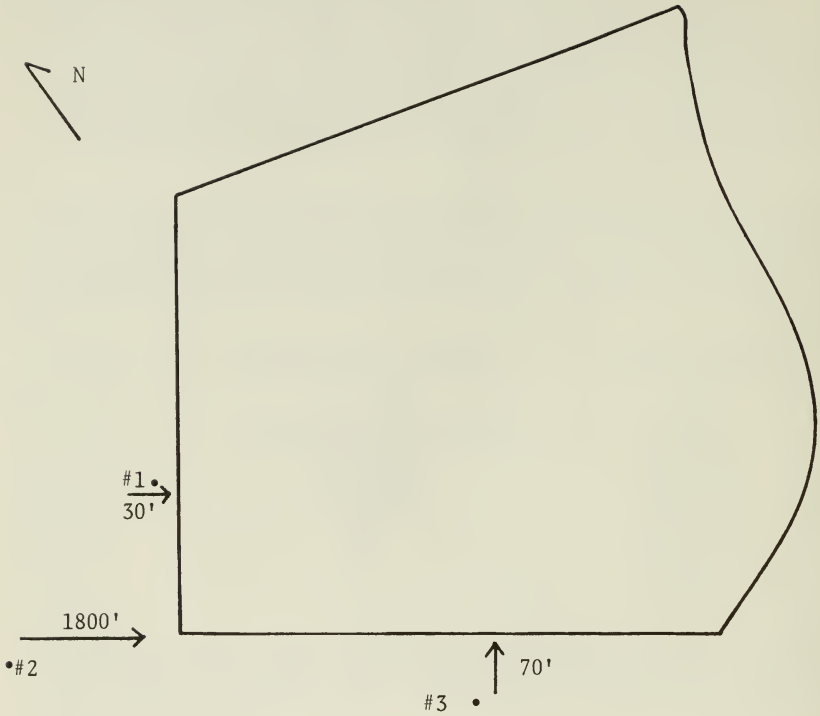
The present monitoring network at the landfill is set up specifically for the Phase I section. As the landfilling activity expands to Phase II and III, more monitoring wells will have to be installed. It is assumed that each phase will have three downgradient wells specific to its area.

3.2 *Case Study*. The expected value of groundwater quality data can be determined by examining the decisions likely to be made with and without sample information. As previously discussed, the Amherst landfill meets the DEQE's proposed monitoring requirements of three downgradient wells. Figure 3 shows the location of the downgradient monitoring wells at the Amherst landfill, with their distance from the perimeter of Phase I and the boundary of the facility. In order to determine the value of information of changing from a two downgradient to a three downgradient well system, a hypothetical two well system must be established. Wells #1 and #2 were chosen to represent the two well system.

Before the empirical application can be performed, the components of Bayesian analysis must be defined for the Amherst landfill. Recall

Figure 3

Phase I Site



1'' = 150 feet

that Bayesian analysis involves states of nature, actions, consequences, and prior and conditional information. These terms are defined seriatim.

3.2.1 *States of Nature.* The difficulty that a decision maker may have in selecting a management action is due to the fact that the true state of nature is unknown. In a hydrologic context, a state of nature is usually defined by a set of given water quality parameters (see McAniff and Willis, 1976). The states of nature used in this research, although dependent on the concentration levels of different parameters, are more complex. Here the states of nature are defined by three characteristics: size of the plume of leachate contaminated water; the parameters or constituents comprising the leachate; and the concentration of the parameters found in the contaminated groundwater. The first category is the breakdown by plume size. Three plume sizes were considered:

(1) no plume, or no leachate entering the groundwater system;

(2) a small plume that extends as far as the nearest well (well #1 for the two downgradient well system, and wells #1 and #3 for the three downgradient well system) but not to the farthest well (well #2);

(3) a large plume that extends at least as far as the most distant monitoring well from the facility perimeter (well #2).

Each plume size is then broken down into two categories which describe the parameters or constituents comprising the leachate. The two categories are sanitary constituents and heavy metal constituents. Those parameters belonging to the sanitary category are generally easier to treat or deal with in a water quality context than those belonging to the heavy metal group. For example, parameters such as pH or iron can be treated by municipal water treatment system. Whereas, heavy metals such as lead or cadmium need special treatment techniques.

Once the parameters or constituents found in the water samples are placed in either the sanitary or heavy metal grouping, further classification by relative concentrations can be made. After a monitoring sample is collected, if the constituents found in the water are sufficiently concentrated to indicate that a leachate problem exists, but the concentration is not high enough to violate any established EPA limits, then the concentration of that parameter is classified as "low". However, if the concentration of the parameter is greater than the EPA limits then the classification is "high".

The reasoning behind the categorization by plume size, constituents and concentration is based on the difference in remedial actions that might be taken dependent on plume characteristics. In other words, the appropriate action for a landfill operator to take in the case of a small plume made up of sanitary constituents in low concentration is going to be different from the appropriate remedial action taken for a large plume composed of heavy metals in high concentration. The actual states of nature are defined as follows:

S_1 = no leachate entering the groundwater;
S_2 = small plume of leachate consisting of sanitary constituents in low concentrations;

S_3 = small plume of leachate consisting of heavy metals in low concentrations;
S_4 = small plume of leachate consisting of sanitary constituents in high concentrations;
S_5 = small plume of leachate consisting of heavy metals in high concentrations;
S_6 = large plume of leachate consisting of sanitary constituents in low concentrations;
S_7 = large plume of leachate consisting of heavy metals in low concentrations;
S_8 = large plume of leachate consisting of sanitary constituents in high concentrations;
S_9 = large plume of leachate consisting of heavy metals in high concentrations.

At this point, the question might arise as to why the joint presence of both sanitary and heavy metal constituents was not included in the classification of the states of nature. In developing the states of nature, it had to be determined whether the existence of one category (sanitary/heavy metal, low/high) was dominant over the other. For example, consider the state of nature S_5 where heavy metals are found in high concentration in water samples. The existence of a leachate plume containing high concentrations of heavy metals is assumed to demand that an action be taken. It is further assumed that because the leachate is made up of heavy metals, the action taken is dominant over any action that would be taken for the existence of sanitary constituents, whether high or low. However, the case for the assumption of a dominant action may break down when a plume made up of sanitary constituents in high concentration exists but heavy metals are also found to be present in low concentrations. It may be argued that if low concentrations of heavy metals are found, then actions will be taken in relation only to those parameters, separate from actions taken with regard to the sanitary constituents. But, for this study, it is assumed that the remedial action concerning the high concentration of sanitary constituents will have to be taken, and is dominant to and inclusive of any action that might be taken concerning the heavy metals. The same reasoning is applied to only one state occurring at a time for this model.

3.2.2 *Actions*. The set of management actions available to an operator of a landfill contains several possibilities for a given situation. However, an action is only admissible if there is no other action which dominates it. Thus, by eliminating inferior actions, the decision maker faces a well-defined set. For the analysis done here, the set of actions must cover the range of states of nature from no leachate to high concentrations of heavy metals.

The hydrogeology of the area must also be considered since the remedial actions taken for a contamination problem are dependent on the underlying system. Methods of control or containment for a leachate plume and their effectiveness will differ in a confined aquifer system relative to a shallow unconfined aquifer. Costs of the relevant actions are also affected by site-specific conditions. In the case of the Amherst landfill, the aquifer that would be affected by leachate is a shallow unconfined aquifer. Hence, the remedial actions considered are those that are appropriate for that type of groundwater system.

The choice of actions also depends on the type of contaminants entering the groundwater. One remedial action may be appropriate for low concentrations of sanitary constituents, while another may be best for high concentrations of heavy metals. However, this does not mean that one action may not be appropriate for more than one state of nature. In some cases, the actions may be the same for two different states. The definition of the six management actions chosen for this study will be preceded by a discussion of the larger set of possible actions. This description includes some of the limitations of each action.

The first and most obvious member of the set of actions is that of taking no action. In the case of no leachate entering the groundwater, this action is appropriate. But, in other situations, some other action is needed.

Next are the actions relating to the identification and study of a contamination problem. Examples of these are increased sampling frequency of groundwater and a greater number of monitoring wells. For example, instead of sampling every six months, the sampling may be increased to every month to watch the movement of a plume. Or, more monitoring wells may be installed so as to gain knowledge on the actual size of a plume.

Other actions involve techniques for the containment and/or control of polluted groundwater (for a more in-depth description see Quince and Gardner, 1982, or Josephson, 1980, or Lehr and Nielsen, 1982). These can be broken down into three categories. Physical containment measures comprise the first category, and include slurry trench cutoff walls, grout curtains, sheet pilings, and hydrodynamic control. Another category includes aquifer rehabilitation methods such as withdrawal, treatment and reinjection, or injection of chemical or biological fixatives. Lastly, a recovery technique may be used to pump up contaminated water for treatment at a municipal sewage treatment facility or treatment for use as a supply source.

The physical containment techniques are a group of methods designed to stop the flow of contaminated groundwater. The object of their use is to control the spread of a plume, physically to restrain its growth in the direction of points of groundwater withdrawal. Slurry trench cutoff walls are constructed by building underground dikes which encircle the leading edge of a plume, hence sealing it off from the rest of the aquifer. Grout curtains are constructed by injecting a cement mixture into subsurface soils to fill the voids between grains. As a result, groundwater flow is restricted. Grout curtains can serve as an effective barrier; however, the degree of impermeability of a curtain is never certain. Sheet piling barriers are interlocking steel sheet piles driven through the unsaturated zone and aquifer to underlying impermeable strata. Problems in the use of these barriers exist if the piles must be driven through coarse, dense material. All of the above have drawbacks in their use. The cost of installation is generally high (for example see Josephson, 1980, p. 1034) ranging from $500,000 upward. Also, their use is limited to shallow aquifers where feasible. But even more important, as stated by James Geraghty, all artificial barriers "eventually leak", hence these techniques are not permanent solutions (Josephson, 1980, p. 1032).

Hydrodynamic control consists of placing a line of pumping wells upgradient from the plume of contaminated water and a line of recharge injection wells downgradient from the plume. Thus, the flow of groundwater

between pumping and injection is stopped or even reversed relative to the original flow direction of the water. The plume is therefore held in check as water is diverted around it. But this type of system demands extensive knowledge of the hydrogeology. Extensive monitoring is also needed to detect any failure in the system. Although hydrodynamic control systems are generally less costly to install than other containment techniques, the operation, maintenance and monitoring costs may be quite large.

Aquifer rehabilitation involves methods of restoring contaminated groundwater to an uncontaminated state. The most common method used is withdrawal, treatment, and reinjection. A series of pumping wells is installed at the head of a plume which brings the contaminated water to the surface for treatment. Following treatment, the uncontaminated water is recharged back into the aquifer either through reinjection or surface recharge. This method is most effective in aquifers with permeable, coarse-grained deposits. However, costs for this type of system are typically very high. Additionally, the line of pumping wells may not intercept the entire plume of contaminated water and where two-phase liquid contamination problems exist pumping may not be feasible. Another technique of aquifer rehabilitation involves injecting some kind of fixative, either chemical or biological, depending on the actual contaminant, into the aquifer itself. Several problems arise with the use of this method. Complete mixing of the treatment with the polluted water may not occur. Further, it is difficult to determine how much fixative should be added to neutralize the water, making sure that the amount is not so much as to create its own contamination problem.

The practice of installing a recovery point system is similar to the aquifer rehabilitation method of withdrawal, treatment, and reinjection. Recovery systems are designed to be used in shallow, unconfined aquifers for the removal of contaminated water. Groundwater is pumped up, and may be treated either by municipal sewage facilities and disposed of or may be treated for use as a supply source. The system may employ a series of well points located perpendicular to the direction of contaminant migration, or may consist of just one or two wells. The drawbacks in the use of this method are similar to those corresponding to other well-using techniques. The recovery wells must be located in the appropriate horizon of the underlying system. In the case of two-phase liquids the method may be infeasible. Lastly, whether all of the contaminated water is recovered or not depends on the size of the problem; it may take many years to cleanse an aquifer, even if the source has been eliminated.

Treatment of contaminated water is a key factor in both recovery well systems and aquifer rehabilitation methods. The ease of treatment of polluted water depends upon the types of contaminants in the water. The treatment needed may be relatively simple as in the removal of a single contaminant, or it may be much more complex. Three classes of treatment methods exist, and each is applicable to different parameters. The classes are physical methods such as adsorption, chemical methods which neutralize some parameters, and biological methods which employ microbiological activity to convert toxics into non-toxic byproducts. Treatment costs are variable and depend on the type and concentration of the contaminants. The quantity of water to be treated and the "acceptable" quality for use are also important factors in cost.

The last action considered, which is the most severe and also most commonly taken action, is abandonment of the resource as a supply. However, this action may not be an alternative in some situations because an unaffected source may not be available, or the costs of developing a new source may be prohibitive. In cases in which a neighboring community may be able to provide the new source, it may not be able to meet present demands fully, or be politically feasible.

The list of possible actions presented above is by no means exhaustive of all the options available. However, the list is representative of the main techniques being applied to problems and discussed in the literature at present. Of course, a landfill manager may want to use a combination of techniques to deal with a contamination problem. But the overall set of management actions open to an operator is limited given the type of leachate problem and the hydrogeology of a site.

In choosing the actions to be used in this analysis, the relative costs and effectiveness were main factors. The actions also had to be those appropriate for a shallow aquifer system. Equally important, the actions had to be those that would be used for the alternative states of nature hypothesized. The actions considered are:

a_1 = no action taken;
a_2 = increase the frequency of water sampling, to once a month;
a_3 = add two additional monitoring wells and increase frequency of sampling to once a month;
a_4 = install a recovery well system with use of the municipal sewage treatment system;
a_5 = install a recovery well system with transportation to a specialized off-site treatment facility;
a_6 = abandonment of wells and establishment of an alternative supply.

Aside from the appropriateness of these actions for a given state of nature, the actions selected were deemed from a cost standpoint to be dominant over the others. Table 7 presents the costs of each action assumed in this analysis. The increased frequency and increased monitoring activities were believed to have the most desirable consequences when the concentrations of constituents found in a plume were low (Josephson, 1982, p. 504A). The recovery well system was considered best when the plume was small but concentrations were high, and also for a large plume of sanitary constituents of high concentration (given as a technique used now in the state by Tom Coucher, 1983). Abandonment was only considered for the state described by a large plume of high concentrations of heavy metals.

3.2.3 *Consequences*. The consequences of any given action will depend on the state of nature that occurs. In some cases, the consequence of an action/state pair will be the capital, operating, and maintenance costs of the action taken. However, in other cases where the action taken is not appropriate for the state occurring, the consequences will include more than just these costs. For example, if a large plume of high concentration sanitary constituents obtains and no action is taken, in the next period the plume may be too large for some remedial action to be taken and abandonment of the supply source may be necessary. Furthermore, the plume may have already affected the supply source. Hence, the consequences or costs in this situation are quite high.

Table 7

Base Costs of Actions

Action		Total Cost
a_1 = take no action		$0
a_2 = increased frequency of sampling to once a month*	$900 per sample	$21,600
a_3 = two additional monitoring wells installed with increased sampling frequency	$900 per sample $2,500 per well	$41,000
a_4 = installation of three recovery wells and use of municipal sewage system for treatment	$12,500 per well $20,000 site study $38,500 sewage use	$95,000
a_5 = installation of three recovery wells and transportation to off-site treatment facilities	$12,500 per well $20,000 site study $112,500 off-site	$170,000
a_6 = abandonment of the supply source and establishment of an alternative supply		$1,000,000

*The sampling cost per well per year at the Amherst facility was given as $1,800. Sampling was done twice a year -- hence, this represents an average cost of $900 per sample. For the base case of a two well system, monthly sampling would cost $21,600 per year.

For the consequences of an inappropriate action given the state of nature occurring, some assumptions on the outcomes or results had to be made. In several cases, the assumption is that in the next period appropriate action will be taken. In the case of a small plume of sanitary constituents of high concentration being the true state of nature, if the appropriate action of installing the recovery well system is not taken in this period, then in the next it is assumed to be. Contrastingly, for other cases, the only appropriate action in the next period is to find an alternative supply. The reasoning behind each of the consequences of the action/state pairs is presented below. The matrix representing these consequences is presented in Table 8.

If action a_1, that of doing nothing, is taken and S_1, having no leachate problem is the true state of nature, then the cost is zero. However, if the state obtaining is S_2, the consequence of not taking the assumed appropriate action of increased sampling is the cost of increased

Table 8

Consequences*

States	Actions					
(S_i)	a_1	a_2	a_3	a_4	a_5	a_6
S_1	0	21.1	40.5	95	170	1000
S_2	21.6	19.1	38.5	95	170	1000
S_3	41	62.6	37.5	150	170	1000
S_4	95	123.2	116	95	170	1000
S_5	1000	1021.6	1041	1150	170	1000
S_6	21.6	19.1	38.5	95	170	1000
S_7	41	62.6	37.5	150	170	1000
S_8	1100	1121.6	1141	95	170	1000
S_9	1200	1221.6	1241	1350	1370	1000

*Consequences are expressed in thousands of dollars.

sampling in the next period. Likewise, if S_3 occurs, the cost of taking a_1 is that of increased monitoring next period. The consequence for the $S_4|a_1$ interaction is similar in that the cost for not acting upon the small plume of high concentration sanitary constituents is the future cost of the recovery well with use of the municipal sewer system. The consequence for S_5 when a_1 is taken is based on the assumption that, if in this period no action is taken on a small plume of high concentration heavy metals, then in the next period the plume will be classified as large and abandonment will take place. The consequences for S_6 and S_7 are identical to those of S_2 and S_3, respectively. The assumptions behind the costs for either S_8 or S_9 being the true state of nature and taking action a_1 are based on the need for abandonment in the next period (since plume size is now too large for recovery in each case) plus an extra cost of damages caused as a result of inaction. These damages may be related to increased treatment costs at the municipal water treatment plant or they may be damages to the health of those consuming water with undetected leachate.

Now consider that the action taken is a_2, with a total cost of \$21,600. Although there is no need for the increased sampling in the case of S_1 occurring, once the action is taken benefits result from increased knowledge about groundwater quality. These benefits can be considered to decrease the net cost of the action. For S_1 being the true state of water quality, the benefits from increased knowledge are assumed to be \$500.00. The idea of benefits from improved knowledge of the groundwater system, and perhaps earlier prediction of a quality problem, extends to S_2 and S_3 also. In the case of S_2, the action is not only appropriate but also yields a \$2,500 reduction in costs due to informational benefits. Although a_2 is not the appropriate action given S_3 occurring, and in the next period increased monitoring is assumed to take place, benefits of \$3,500 in identification of the heavy metal problem are presumed to result. Similar to the reasoning behind the consequence given S_2, if S_4 existed a \$2,500 information gain would occur, but the

action in the next period would be installation of a recovery well system (a_4), and this cost must be added to the cost of increased sampling. Given a small plume of high concentration of heavy metals S_5, the consequence is similar to the $S_5|a_1$ cost but here the costs of sampling are added. The costs for S_6 and S_7 are based on the same reasoning as S_2 and S_3, respectively. Finally, as in the case of choosing a_1, when S_8 or S_9 occur, the consequences are assumed to be those of abandonment and damages, now with sampling costs included.

Adding two more downgradient monitoring wells and increasing sampling frequency characterize action a_3, with a base cost of $41,000. Information benefits resulting from an action apply in this case also. These benefits are the same amounts assumed for action a_2 -- $500, $2,500 and $3,500, respectively, for S_1, S_2 and S_3. Hence, these benefits are subtracted from the base cost of the action for these three states of nature. The consequence resulting from the $S_4|a_3$ interaction is the cost of a_3 plus next period's cost of installing a recovery well system (a_4). However, because the additional wells may serve the same purpose as the limited site-investigation these costs are subtracted from the total. Once again the cost for the true state being S_5 is that of abandonment in the next period plus the costs of the action taken in the current period. Also, S_6 and S_7 have the same consequence as when S_2 or S_3 occurs. Similarly, the consequences for S_8 and S_9 follow the reasoning given for both a_1 and a_2.

The consequences of choosing a_4 are the costs of pumping groundwater from the aquifer to the municipal sewer system, assumed to be $95,000 in this case. If a plume of high concentration sanitary constituents exists (S_4 and S_8), this is the appropriate action. Although the action is not appropriate for S_1, S_2 or S_6, there are also no adverse effects from taking it. Therefore, for the occurrence of one of these three states, the cost is that of the recovery well system. However, if the true state of nature is either S_3 or S_7, pertaining to heavy metals in low concentration, then there is assumed to be some damage to the municipal sewage treatment plant. Consequently, this damage cost is added to the cost of the recovery system. But, if S_5 or S_9 occurs, then the consequence is equal to the cost of the recovery system, damages to the plant, and abandonment in the next period. For the large plume, S_9, the additional damages due to health costs or increased treatment at the supply source are included.

The consequences of undertaking action a_5, at a cost of $170,000, are the same for all of the states of nature except S_9. For S_1 through S_8, the costs resulting from a_5 are those of recovering the contaminated groundwater (uncontaminated in the case of S_1) and then treating it as if it contained heavy metals. However, when S_9 is encountered, the plume is assumed to be too large for this action effectively to recover all of the contamination. Therefore, abandonment will take place in the next period. Thus, the costs for this $S_9|a_5$ interaction are abandonment plus the remedial action costs.

Finally, for any state occurring, the costs of abandonment (a_6) remain the same -- $1,000,000 for development of an alternative supply. The reasoning behind this assumption is that, although there is loss of the resource in all cases, it is assumed that there are no other effects of the action. Hence, whether the action is appropriate or not, its costs are invariant.

3.2.4 *Prior Probabilities*. The prior probabilities are derived from the knowledge that an operator has about the state of nature before any experimental sampling is performed. The prior probabilities may be formed from either data-based or non-data-based sources. For this analysis, the plume sizes, characteristic constituents, and concentration levels are regarded as discrete random variables and, hence, are subject to a discrete probability distribution. Three sets of prior probabilities were used in the analysis. The probabilities were established on the basis of present belief regarding water quality parameters at the landfill.

Because the states of nature are comprised of three components, the formulation of the priors had to be accomplished in three steps. The sum of the probabilities across each category (i.e., plume sizes) had to equal one. Within each plume size, the sum of probabilities of either sanitary constituents or heavy metals occurring in the leachate had to be one. Lastly, the sum of the probabilities on high or low levels had to equal one. Thus, for each set of priors the sum is one.

Because past water sampling has not indicated any leachate problem, the prior probability of S_1, the state of no leachate entering the system is quite high. However, because leachate is now being collected on the landfill liner surface the probability is not negligible. For probability schemes I and II, the likelihood of sanitary relative to heavy metals problems and low relative to high levels were considered equal. But, for scheme III, the likelihood of a plume being characterized by sanitary constituents was given greater weight. The three probability sets are presented in Table 9.

3.2.5 *Conditional Probabilities*. The prior knowledge of the state of groundwater quality can be refined or improved through the use of monitoring wells. Sampling from the monitoring network can provide landfill operators with more information on water quality. This additional information takes the form of predictors (Z_k) of water quality parameters. Different monitoring networks will provide different results, some more precise in predicting the true state of water quality than others. The precision level of a network, expressed conditionally on the states of nature, constitutes a sampling distribution for the estimators. The predictors of S_i used here are:

Z_1 = no leachates detected;
Z_2 = well #1 detected sanitary constituents in low concentration, well #2 did not;
Z_3 = well #1 detected heavy metal constituents in low concentration, well #2 did not;
Z_4 = well #1 detected sanitary constituents in high concentration, well #2 did not;
Z_5 = well #1 detected heavy metal constituents in high concentration, well #2 did not;
Z_6 = well #2 detected sanitary constituents in low concentration;
Z_7 = well #2 detected heavy metal constituents in low concentration;
Z_8 = well #2 detected sanitary constituents in high concentration;
Z_9 = well #2 detected heavy metal constituents in high concentration.

The above Z_k's are for the two downgradient well system. When dealing with the three downgradient well system, Z_2 through Z_5 represent sampling for both well #1 and well #3. Recall that well #1 and well #2 give the

Table 9

Prior Probabilities

Prior Set	State	Probabilities
I	S_1	.85
	S_2	.025
	S_3	.025
	S_4	.025
	S_5	.025
	S_6	.0125
	S_7	.0125
	S_8	.0125
	S_9	.0125
II	S_1	.90
	S_2	.0175
	S_3	.0175
	S_4	.0175
	S_5	.0175
	S_6	.0075
	S_7	.0075
	S_8	.0075
	S_9	.0075
III	S_1	.90
	S_2	.021
	S_3	.014
	S_4	.021
	S_5	.014
	S_6	.009
	S_7	.006
	S_8	.009
	S_9	.006

precision for the two well system and that the three well precision level corresponds to all three downgradient wells.

Consider a monitoring network which has a precision level of U*. Associated with this network is a corresponding set of conditional probabilities, $P(Z_k|S_i)$. As background for developing these conditional probabilities, several remarks are in order. First, states S_1 through S_9 represent an ordering from best to worst case. Results Z_1 through Z_9 are their corresponding predictors (not always accurate, of course). Therefore, as the monitoring system becomes more accurate (more wells, more frequent monitoring), this increased accuracy is reflected in larger $P(Z_k|S_i)$ for k=i and smaller $P(Z_k|S_i)$ for $k \neq i$. If the monitoring system were perfect, the $P(Z_k|S_i)$ could be arranged into an identity matrix of order 9.

Since the data were unavailable, it was necessary to construct the matrix of conditional probabilities based on expert judgment and a set of

assumptions. The expert judgment involved eliciting estimates of the accuracy of a particular monitoring system. The experts were hesitant to respond with probability estimates, and perhaps had a difficult time formulating their experience in these terms. (See Willis and Perlack [1980] for a partial survey of literature indicating the limited information processing ability of the human mind and in particular to the formulation of uncertain events in a probabilistic context. See also Lin, Dean and Moore [1974] in this latter area.) When the experts did respond with a probability estimate, it seemed clear that this probability estimate (P) applied to larger aggregates of states than the nine envisioned here. It is assumed that the states are viewed as three groups: S_1, S_2 through S_5, and S_6 through S_9; i.e., no plume, small plume, and large plume. Thus, their responses could be interpreted as $P(Z_1|S_1)$; $P(Z_2, Z_3, Z_4$ or Z_5 given S_i, $i=2,\ldots,5)$; $P(Z_6, Z_7, Z_8$ or Z_9 given S_i, $i=6,\ldots,9)$. Finally, it is assumed that $P(Z_k|S_{i+1})$ declines with increases in the size of j. The standard normal density function was used to approximate this decline.

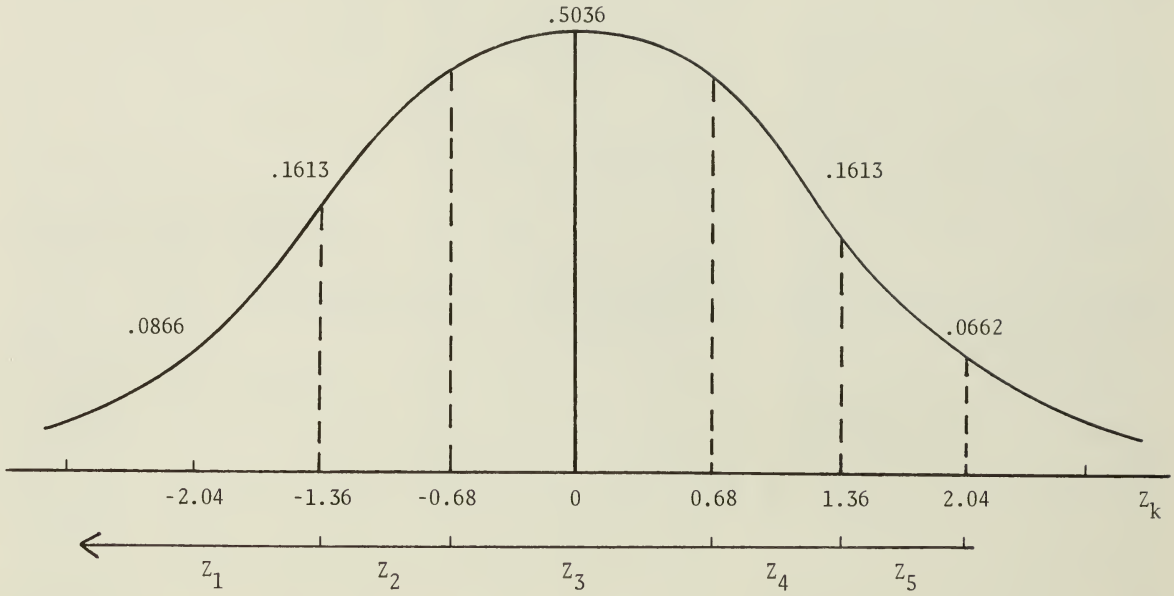
For purposes of illustration, the procedure used in the formulation of the conditional probabilities for this model are presented using the 90 percent precision level given by one expert. (Two estimates of precision levels of 90 and 85 percent for the Amherst landfill's monitoring network were given, respectively, by engineers Tom Coucher and Peter Mockrzecky.) First, recall that all of the states of nature fall within one of the three plume sizes; no plume, small plume, and large plume. Also, recall that each of the plume sizes are broken into four characterizations (except for no plume). Because each characterization is a discrete random variable, the probabilities are discrete in form. However, within each plume size the states of nature are considered to be ordered. As a result, the probability of the sample state of nature Z_k being the same as the true state of nature S_i is distributed among the four possibilities within a plume size. An illustration of this distribution is presented in Figure 4.

In this case, the true state of groundwater quality is represented by S_3. By constructing a confidence interval around S_3, the probability of prediction by sample Z_k can be found. The interval around a state can be made as large or small as needed depending upon the precision level. For this example, the area under the curve represented by Z_{3+1} and Z_{3-1} is equal to .5036. Thus, the probability of observing Z_3 given S_3 is equal to .5036. The remaining 39.64 percent of the 90 percent confidence level is represented by the probabilities of observing Z_2, Z_4 and Z_5 given S_3 occurring. Hence, the 90 percent precision level is distributed among the four small plume alternatives. The interpretation, for example, of the probability of Z_5 resulting, given the true state of nature is S_3, is that there is a 6.62 percent chance that the sample will predict heavy metals in high concentration when in reality they exist in low concentration. Since Z_2 through Z_5 account for 90 percent of the area beneath the curve, the remaining 10 percent of the area is accounted for by the other five Z_k.

The above example presents one of the precision levels for the three downgradient well system analyzed here. One other precision level of 85 percent was considered for this system. Three different levels were considered for the two well system; 80, 70 and 60 percent. By using these ranges, the changes in value of information can be compared with respect to relative changes in monitoring precision. As indicated, the two precision levels of 90 and 85 percent were estimates of the Amherst landfill's

Figure 4

Monitoring Precision: $P(Z_k|S_3)$



monitoring network precision given by two engineers (Mockrzecky, 1983, and Coucher, 1983). The range of 80 to 60 percent chosen for the two well system was established to represent the change in precision between a three and a two well system, from little change to a large change. The matrices of conditional probabilities representing the varying precision levels are provided in the Appendix.

4. Empirical Results

The results of the case study application to Amherst's situation are provided in the section immediately below. In an effort to generalize the results somewhat, to extend to other situations in which it is believed that the likelihood of groundwater contamination is higher, we provide a further set of simulations in the subsequent section.

4.1 *Amherst Application.* The expected value of information is determined by the difference in the expected value of the decision made with and without the sample data. Since this analysis included three sets of priors and five sets of conditional probabilities, there are fifteen possible outcomes to the problem. Table 10 summarizes the results, presenting the expected value of information for all fifteen simulations and also for the case of perfect information.

Table 10

Expected Value of Information*

Monitoring System	Prior Set I	Prior Set II	Prior Set III	Precision Level
1	21.0941	13.8675	13.4072	90
2	19.3709	12.4632	12.0099	85
3	16.8129	10.1195	9.74306	80
4	12.1507	5.6825	6.10395	70
5	6.6888	3.4833	4.11298	60
EVPI	36.0375	23.7125	22.01	

*Expressed in thousands of dollars.

The action having the minimum expected value given only the prior information was a_1, that of taking no action, in all three scenarios. The three expected values were $58,472.50, $37,977.50 and $34,563.00, respectively, for prior sets I, II and III (prior sets defined in Table 9). For the Bayesian strategy, the only actions selected as minimizing expected value given the varying Z_k's were a_1, a_4 and a_5. Recall that a_4 was installing a recovery well system and using the municipal sewage system for treatment, and that a_5 was also installing a recovery well system but having transport of wastes to an off-site treatment facility (Section 3.2.2). Among these actions, the one chosen for a particular Z_k result did not vary much throughout the 15 solutions. Only in the case of perfect sample information on the true state of water quality were all six actions chosen. Of course, the actions chosen, given perfect information, were those assumed to be appropriate originally.

At this point it may be instructive to analyze in detail one set of results. The results based on a 90 percent precision level for the three downgradient well monitoring network, the most optimistic precision

estimate given by experts, and prior set I are presented below. A general overview of the other results will follow.

Given the 90 percent precision level and prior set I, characterized by $P(S_1) = .85$ and equal weighting between sanitary relative to heavy metal constituents and low relative to high levels, the optimal action taken for the sampling results of Z_1 through Z_4 was a_1. The reason for this is due, partly, to the high prior for $P(S_1)$ as compared to $P(S_2)$ through $P(S_4)$ -- .85 compared with .025. Further, for each of these cases, the penalty for taking action a_1, doing nothing, instead of the appropriate action (a_2, a_3 or a_4) is relatively small. For example, in the case of taking a_1 instead of a_2 which is increased sampling frequency, when S_2, a small plume of low concentration sanitary constituents, is the true state of nature the penalty or cost is only an additional $2,500. Thus, given the posterior probabilities, the expected value is smaller for action a_1. The four expected values for these sample results are: $Z_1|a_1 = \$545.78$, $Z_2|a_1 = \$6,563.70$, $Z_3|a_1 = \$33,635.60$, $Z_4|a_1 = \$145,066.70$. The action chosen as minimizing expected values for sample results of Z_5 through Z_7 was a_5. For result Z_5, indicating a small plume of high concentration heavy metals, this is the appropriate action. However, for results Z_6 and Z_7, a large plume of low concentration sanitary or heavy metal constituents, respectively, it is not. In these cases, because the only consequence of taking a_5, installing a recovery well system with off-site treatment, is the cost of action a_5 itself with no additional costs, the expected value is less than for the other actions which have large consequences for the action taken being inappropriate. The expected values are $Z_5|a_5 = \$171,848.00$, $Z_6|a_5 = \$186,236.70$, and $Z_7|a_5 = \$24,549.50$. Given the sample result being either Z_8 or Z_9, a large plume of high concentration sanitary or heavy metal constituents, respectively, the optimal action was a_4, yielding expected values of $360,127.50 and $952,559.00.

For this scenario, the $P(Z_k)$'s or the probabilities of a sample result occurring were, respectively, 0.64798875, 0.15585125, 0.078835, 0.0379275, 0.02435, 0.0157975, 0.01332, 0.011665 and 0.01392375. The expected value of the Bayesian strategy, therefore, is $37,378.40. Thus, the expected value of information in this case, W in the first row and column of Table 10, is $21,094.10 -- i.e., $58,472.50 less $37,378.40.

Overall, for the three cases in which the precision of the monitoring network is at a 90 percent level, the optimal action for sample results Z_1 through Z_4 is a_1, that of do nothing. For results Z_5 through Z_7, indicating the states of nature of a small plume of high concentration heavy metal constituents, a large plume of low concentration sanitary constituents, and a large plume of low concentration heavy metal constituents, respectively, the optimal action is a_5. For results Z_8 and Z_9 indicating a large plume of high concentration sanitary or heavy metal constituents, respectively, the optimal action is a_4.

At an 85 percent precision level, using prior sets I and II, the optimal action given Z_8 occurring becomes a_5. However, for the model using prior set III (more weighting on the probability of sanitary constituents occurring than heavy metals occurring), the optimal action for Z_8 remains a_4 at this precision level.

The actions minimizing expected value at the 80 percent precision level are the same as those at the 85 percent level. But, now for the case using prior set III, the optimal action for Z_8 becomes a_5.

For the 70 percent precision level considered representative of the two downgradient well system, once again the models using prior sets I and II differed in optimal actions from the model using set III. In the two first cases, Z_1 through Z_4 continue to indicate action a_1 as that action minimizing expected value. Action a_5 is the optimal action for sample results Z_5 to Z_9, inclusive. However, for the model using prior set III, the action minimizing expected value for example results Z_1 through Z_4 is still a_1, and a_1 is also now the optimal action for Z_5 and Z_6, while a_5 is for the remainder.

The lowest precision level considered in the case study was that of sixty percent. For the model using prior set I, a_1 was the optimal action for Z_1 through Z_6 (similar to the 70 percent level using prior set III). The optimal action for the remaining possible sample results, Z_7, Z_8 and Z_9 was a_5. Hence, the two dominant actions throughout the 15 scenarios are a_1 and a_5. The action a_4 only comes into effect at the 90 and 85 percent precision levels.

Recall that the 90 and 85 percent levels represent estimates of precision of the three downgradient well system given by two engineers. By looking at rows one and two of Table 10, a comparison between the value of information for each precision level can be made. Consider rows one and two of column one, models using the prior set I. The difference in the value of information between the two is about $1,725. Similarly, the difference between the value of information for the two precision levels given by prior sets II and III is about $1,400. These differences seem sufficiently slight so as to make it of little consequence to determine whether the true precision level is 85 or 90 percent. However, as will be seen shortly, these differences in the value of information between the precision levels will be important in comparison of the three downgradient well system with the hypothesized precision levels of the two well system.

Before comparing the three well system to the two well system, knowledge can be gained from the difference in the value of information figures for the two alternative three well precision levels. Referring back to the 90 and 85 percent levels for prior set I, the difference was $1,725. Now if we assume the 85 percent level to be the correct precision level at the Amherst facility, the possible decision to add another downgradient well (the fourth), raising the precision level to 90 percent, can be analyzed. If the cost of the fourth well is greater than $1,725, as it is likely to be since well costs were quoted at approximately $2,500, then from an economic efficiency point of view the fourth well should not be installed. Although the value of information from the 90 percent precision level is still greater than the costs (approximately $10,000) of obtaining that precision level, the marginal value of the fourth well is not greater than its marginal costs. (This is similar to the example presented in Figure 1. Here, the 90 percent level could be represented by the monitoring level of U_1, still feasible but beyond the point of optimality.) The same conclusions result using either prior set II or III as the base model.

The prior probability sets used in the model were chosen to represent two levels of knowledge about the present groundwater quality at the landfill site. Probabilities of 85 and 90 percent were assigned to the likelihood that S_1, no leachate plume, was the true state of nature. By comparing the value of information figures under these two prior levels, the value of monitoring given differing accuracies of prior knowledge can be

determined. For example, the value of information at the 90 percent precision level for prior set I, $P(S_1) = .85$, is equal to about $21,000. The value of information for the same precision level, but under prior set II, $P(S_1) = .90$ (set II assumed in order to compare systems with similar weightings for sanitary relative to heavy metal constituents and low relative to high concentrations), is equal to $14,000. The difference in the value of information due only to the variation in the prior probabilities is then equal to approximately $7,000. In general, the accuracy of the prior knowledge has much to say about the value of monitoring -- if a lot is known about groundwater quality before monitoring is undertaken, the monitoring has relatively little informational value.

The levels used in this analysis to describe the precision of a two downgradient well system were 80, 70 and 60 percent. These were alternative estimates about the accuracy of the landfill monitoring system without one of the downgradient wells. For illustration, the 70 percent precision level will be compared to the 85 percent precision level for the three well system. At the 70 percent level the value of information (using prior set I) is equal to about $12,000. The difference in the value of information for this level and that of the 85 percent level is the value of information attributed to the third downgradient well. Here, that value is equal to roughly $7,250. Therefore, the value of the third well is much greater than its assumed costs of $2,500. Hence, for this case, adding the third downgradient well is an economically justified action, and is also movement towards the optimal precision level. The same conclusion holds for a comparison between these two levels for the two other models using prior sets II and III. However, the value of information attributed to the third well is less in each case, approximately $6,700 and $6,000, respectively. Clearly, if the value of information of a third well is greater than its costs at a 70 percent precision level, it will also be true for the 60 percent level, having greater value given the 60 percent level. Increasing precision from an 80 percent to an 85 percent level may, however, not be economically justified. In all three models, the benefits gained from the third well would be close to its costs. Thus, although the value of the system at an 85 percent level would still be greater than the costs of the system, the decision would be marginal.

4.2 *Extended Results*. In order to determine whether the results of the Amherst case study could be extended to other situations in which the likelihood of contamination is greater, additional simulations were performed. Each of these simulations used the 90 percent precision level as representative of the three downgradient well system and the 70 percent precision level representing the two well system.

To simulate aquifers with greater odds of being contaminated, the three prior probability sets (I, II and III) were modified successively by reducing $P(S_1)$ by .05 and apportioning this .05 among the remaining states of nature in the same proportion as the original $P(S_i)$. Eight such prior probability sets were generated corresponding with each of the three case study priors. These are provided in Table 11. Note that alternatives 1 through 8 represent situations in which the likelihood of contamination steadily increases.

The analysis performed in the previous section was repeated using these alternative priors (representations of aquifer conditions).

Table 11

Alternative Priors

Case Study Priors	Alternative	$P(S_1)$	$P(S_2)$	$P(S_3)$	$P(S_4)$	$P(S_5)$	$P(S_6)$	$P(S_7)$	$P(S_8)$	$P(S_9)$
	1	.8	.0333	.0333	.0333	.0333	.0166	.0166	.0166	.0166
	2	.75	.0416	.0416	.0416	.0416	.02083	.02083	.02083	.02083
	3	.7	.05	.05	.05	.05	.025	.025	.025	.025
	4	.65	.0583	.0583	.0583	.0583	.02916	.02916	.02916	.02916
I	5	.6	.066	.066	.066	.066	.033	.033	.033	.033
	6	.55	.075	.075	.075	.075	.0375	.0375	.0375	.0375
	7	.5	.083	.083	.083	.083	.0416	.0416	.0416	.0416
	8	.45	.0916	.0916	.0916	.0916	.0458	.0458	.0458	.0458
	1	.85	.0262500	.0262500	.0262500	.0262500	.011250	.011250	.011250	.011250
	2	.8	.035	.035	.035	.035	.015	.015	.015	.015
	3	.75	.04375	.04375	.04375	.04375	.01875	.01875	.01875	.01875
	4	.7	.0525	.0525	.0525	.0525	.0225	.0225	.0225	.0225
II	5	.65	.06125	.06125	.06125	.06125	.02625	.02625	.02625	.02625
	6	.6	.07	.07	.07	.07	.03	.03	.03	.03
	7	.55	.07875	.07875	.07875	.07875	.03375	.03375	.03375	.03375
	8	.5	.0875	.0875	.0875	.0875	.0375	.0375	.0375	.0375
	1	.85	.0315	.021	.0315	.021	.0135	.009	.0135	.009
	2	.8	.042	.028	.042	.028	.018	.012	.018	.012
	3	.75	.0525	.035	.0525	.035	.0225	.015	.0225	.015
	4	.7	.063	.042	.063	.042	.027	.018	.027	.018
III	5	.65	.0735	.049	.0735	.049	.0315	.021	.0315	.021
	6	.6	.084	.056	.084	.056	.036	.024	.036	.024
	7	.55	.0945	.063	.0945	.063	.0405	.027	.0405	.027
	8	.5	.105	.07	.105	.07	.045	.03	.045	.03

Specifically, the EVPI was calculated, along with the expected value of information associated with the two and the three well systems. These results are depicted in Table 12. Consistent with the case study findings, the expected value of information from both systems, as well as for perfect information, increased monotonically as $P(S_1)$ declined. Further, in each case the increase was approximately linearly related to this decline in $P(S_1)$. It is worthy of note that the benefits of information from the two well system grow linearly at a slower rate than do the benefits for the three downgradient well system. Hence the marginal benefits of the third well (last column of Table 12) increase linearly as $P(S_1)$ declines.

Table 12

Value of Information with Alternative Priors*

Prior Set	Alternative	EVPI	3 Well System	2 Well System	Difference
	1	48.05	28.3125	18.6206	9.69
	2	60.0625	35.8853	25.0905	10.79
	3	72.075	43.756	31.5604	12.20
	4	84.0875	51.6266	38.0303	13.60
I	5	96.1	59.4973	44.5003	14.99
	6	108.112	67.3679	50.9702	16.4
	7	120.125	75.2386	57.5342	17.70
	8	132.137	83.1093	64.9622	18.15
	1	35.5688	21.0817	12.1533	8.93
	2	47.425	28.2959	18.624	9.67
	3	59.2813	36.0203	25.0948	10.93
	4	71.1375	43.918	31.5655	12.35
II	5	82.9938	51.8157	38.0363	13.78
	6	94.85	59.7133	44.507	15.21
	7	106.706	67.611	50.9778	16.63
	8	118.563	75.5087	57.7742	17.73
	1	33.015	20.3913	11.541	8.85
	2	44.02	27.3754	17.8077	9.57
	3	55.025	34.3595	24.0744	10.29
	4	66.03	41.3436	30.3411	11.00
III	5	77.035	48.3277	36.6078	11.72
	6	88.04	55.5943	42.8744	12.72
	7	99.045	62.9771	49.1411	13.84
	8	110.05	70.3599	55.4078	14.95

*All values in thousands of dollars.

In each case, the addition of a third downgradient well produces expected benefits well in excess of the likely cost. Thus, for situations similar to our case study, the proposed regulation makes economic sense. Further, if the probabilities of a contaminated aquifer are higher, but all other factors are similar, the third downgradient well is even more justified.

5. Conclusions

The aim of this research was to establish a framework for valuing the information gained from groundwater monitoring at sanitary landfills. The impetus for the analysis was the regulation proposed by DEQE requiring three downgradient monitoring wells at all sanitary landfills in the state. This requirement means increasing the number of downgradient monitoring wells for the majority of landfills in the state, which presently have two downgradient wells. By proposing this requirement, the DEQE has implicitly placed a value on the information produced by the additional downgradient well.

The first objective was to estimate the value of information attributable to the third well. The second objective was to find an economic limit to the amount of monitoring at a landfill facility, using the value of information. In order to accomplish both of these objectives, a methodology for determining the value of information from monitoring networks with varying precision levels had to be established. The methodology utilized was that of a Bayesian theoretic approach.

The sanitary landfill facility in Amherst, Massachusetts was used as a case study for the analysis. Two precision levels of 90 and 85 percent were assumed for the three downgradient well system existing at the facility. A range of precision levels, 80, 70 and 60 percent was chosen to represent the hypothetical two downgradient well system. In analyzing DEQE's proposed regulations, a comparison between the value of information from a precision level for the three downgradient well system with that for the two downgradient system, provides the marginal expected value of information specific to the third well. If the 90 percent and 70 percent precision are assumed to be the actual levels of the two systems, the value of information attributed to the third well is approximately $9,000, $8,000 and $7,000, respectively, for prior sets I, II and III. However, if 85 and 70 percent are the true precision levels, then the value of information gained from the third well is about $7,000 for all three prior sets. In any of these scenarios, the expected value of information is greater than the marginal costs of adding the third well. Hence, the requirement of the third well would be economically justified for these cases. However, if the two precision levels are 85 and 80 percent, the marginal cost of the third well is as great as, or greater than, the value of information gained from its addition to the monitoring network. Thus, the requirement of the third well may not be justified.

Recall that the optimal level of monitoring precision is that level at which the marginal benefits are equal to marginal costs. Therefore, the costs of the last well should be equal to the benefits gained from its information. Unless this is true, either adding another well will have gains greater than its cost or adding another well will have costs greater than its gains. Examples of both situations can be found in this study. For example, if the two downgradient well system has an 85 percent precision level, then increasing the precision to the 90 percent level by adding a third well is not economically justified since the benefits gained from the information from that well are less than the costs of the well. On the other hand, going from a 60 percent precision level to an 85 percent level is economically justified since the gains are greater than the costs.

In considering the overall value of information from monitoring, the value changes over the different prior schemes. For the system with a 90 percent precision level, the value of information, given prior knowledge predicting an 85 percent likelihood of no leachate entering groundwater (prior set I), is equal to roughly $21,000. At the same precision level for the monitoring system, but assuming either of the other prior schemes having a 90 percent likelihood of no leachate entering groundwater (sets II and III), the value of information is roughly $13,500. Given a monitoring network with an 85 percent precision level, the value of information is approximately $19,000 for prior set I, and $12,000 for both prior sets II and III. Hence, the monitoring networks represented by these precision levels yield sampling information of greater value than their assumed costs.

Finding an upper limit on the value of information at the Amherst facility entailed deriving the value of information given perfect knowledge about groundwater quality at the facility. Our findings indicated the expected value of perfect information given the three prior sets was close to $36,000, $24,000 and $22,000 respectively. These values set the upper limit on monitoring costs that should be undertaken. However, these limits should never be reached because the cost of perfect information would be prohibitive.

The values estimated in this study are to be considered illustrative only. Because of the difficulty encountered in obtaining data on costs of remedial actions, the figures used are assumptions concerning the magnitudes of costs that might be realized. With better information on actions that might actually be undertaken given a leachate contamination problem, and more accurate costs of those actions for the specific hydrogeology of a landfill site, the methodology established here could be used to determine the optimal precision level for the landfill facility in question.

One limitation of this analysis was the time frame used in determining costs. Instead of the two period framework, a more appropriate time period may be a much larger number of years. By considering the costs over several years in present value terms, the consequences of the state/action interactions may become more realistic. Also, a better estimate of the value of information gained from a monitoring network precision level for the lifetime of the facility could be provided. Once again, however, in order to estimate the value of information over several time periods, more accurate data than could be obtained for this research would be needed.

Beyond the question of the optimal number of monitoring wells at a sanitary landfill facility, there are several other important considerations. One is the design and siting of the monitoring wells. Presently, debate is ongoing among those familiar with the operation and design of monitoring networks. This debate centers on the question of whether the wells should be located at a landfill area perimeter or the facility boundary. An evaluation of the merit of either position is beyond the scope of this research, but should be seriously considered in the final framing of the proposed requirements. The final proposal should require wells to be sited so as to maximize the informational value resulting from that design of monitoring network.

Additionally, in many situations, increased sampling frequency from wells may provide early detection of a leachate problem. At this time, sampling is only required every six months. But, more frequent sampling may be required if a contamination problem is suspected. From the results of this research, if one assumes increased sampling to result in an increase in the accuracy or precision of a network, then a greater frequency than twice a year may be justified based on the value of information from the sampling. Once the wells have been installed, the marginal cost of each sample is relatively low. The optimal sampling frequency should be determined similarly to the determination of the optimal number of monitoring wells. Based on the results of this research, it seems likely that it is economically justified to increase the frequency of sampling.

Another concern with the proposed regulations is with how the monitoring and other requirements were formulated. During the research for this study, the lack of economic justification became apparent. The number of wells to require should not be determined arbitrarily, but rather should be based on economic criteria. Our analysis suggests the proposed regulations may be quite rational from an economic perspective; but this should not be left to chance. Also, concern over the real objectives of the proposed monitoring requirements developed. In addition to the informational value that monitoring wells could have, they constitute a requirement which would increase the costs of operating a landfill. By increasing these costs, communities would be given an incentive to move toward refuse reuse and recovery plans. This may be a desirable direction. However, from an economic efficiency stance this same incentive could be given in a more efficient manner, such as through taxation. If this is part of the reason for the monitoring proposals, perhaps it should be made explicit.

Unrelated to the actual monitoring question, but equally important, in the overall groundwater protection issue is that of land use. The Aquifer Recharge Protection Program is an existing state program for communities to receive state aid for the purchase of land forming major aquifer recharge zones. Amherst and Belchertown are two towns which have together applied to the state to receive funding. However, the recharge area which these two towns wish to buy is part of the Lawrence Swamp, which is also the location of the municipal sanitary landfill used by the communities. It would seem that the first step in aquifer protection is the consideration of the land use in the area.

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APPENDIX

In Section 3.2.5, it was stated that a set of conditional probabilities can be formulated so as to define the precision level of a given monitoring network. Recall that an example of a monitoring network with a 90 percent precision level was presented. Figure 4 shows the distribution representing this precision level for $P(Z_3|S_3)$. Through examination of this figure, one notices an asymmetry between the areas for Z_1 and Z_5 which are both two standard deviations from Z_5. In defining the conditional probability for Z_1, a complication arises since Z_{1-k} has not been defined. To remedy this problem, it is assumed that Z_1 incorporates all of Z_{1-k}. For this case then, the value of $P(Z_1|S_3)$ is equal to .0866 (.0662 + .0171 + .003 + .0003). This same problem arises for Z_9 in several cases, and is dealt with in the same manner.

The values of the conditional probabilities were determined using the standard normal distribution. As described in Section 3.2.5, the sum of four consecutive $P(Z_k|S_i)$ will be equal to the precision level of a monitoring network for a given state i. The values which define the areas under the normal curve for each conditional probability are as follows: for the 90 percent precision level, z = .68 gave the area for Z_k, k=i, and z = 1.36 gave the area for Z_{k+1}, and z = 2.04 gave the area for Z_{k+2}, etc.; for the 85 percent precision level, z = .6 gave the area for Z_k, k=i, z = 1.2 gave the area for Z_{k+1}, etc.; for the 80 percent precision level, z = .52 gave the area for Z_k; for the 70 percent precision level, z = .44 gave the area for Z_k; for the 60 percent precision level, z = .34 gave the area for Z_k.

The five matrices of conditional probabilities are presented in Tables 13 through 17 below.

Table 13

Conditional Probabilities ($P(Z_k|S_i)$) for U = .90

State of Nature	Sample Estimate Z_k								
S_i	Z_1	Z_2	Z_3	Z_4	Z_5	Z_6	Z_7	Z_8	Z_9
S_1	.7518	.1613	.0662	.0171	.003	.0003	0	0	0
S_2	.2479	.5036	.1613	.0662	.0171	.003	.0003	0	0
S_3	.0866	.1613	.5036	.1613	.0662	.0171	.003	.0003	0
S_4	.0204	.0662	.1613	.5036	.1613	.0662	.0171	.003	.0003
S_5	.0033	.0171	.0662	.1613	.5036	.1613	.0662	.0171	.0033
S_6	.0003	.003	.0171	.0662	.1613	.5036	.1613	.0662	.0204
S_7	0	.0003	.003	.0171	.0662	.1613	.5036	.1613	.0866
S_8	0	0	.0003	.003	.0171	.0662	.1613	.5036	.2479
S_9	0	0	0	.0003	.003	.0171	.0662	.1613	.7518

Table 14

Conditional Probabilities ($P(Z_k|S_i)$) for U = .85

State of Nature	Sample Estimate Z_k								
S_i	Z_1	Z_2	Z_3	Z_4	Z_5	Z_6	Z_7	Z_8	Z_9
S_1	.7258	.1591	.0792	.0277	.0069	.0011	.0002	0	0
S_2	.2742	.4516	.1591	.0792	.0277	.0069	.0011	.0002	0
S_3	.1151	.1591	.4516	.1591	.0792	.0277	.0069	.0011	.0002
S_4	.0359	.0792	.1591	.4516	.1591	.0792	.0277	.0069	.0013
S_5	.0082	.0277	.0792	.1591	.4516	.1591	.0792	.0277	.0082
S_6	.0013	.0069	.0277	.0792	.1591	.4516	.1591	.0792	.0359
S_7	.0002	.0011	.0069	.0277	.0792	.1591	.4516	.1591	.1151
S_8	0	.0002	.0011	.0069	.0277	.0792	.1591	.4516	.2742
S_9	0	0	.0002	.0011	.0069	.0277	.0792	.1591	.7258

Table 15

Conditional Probabilities ($P(Z_k|S_i)$) for U = .80

State of Nature	Sample Estimate Z_k								
S_i	Z_1	Z_2	Z_3	Z_4	Z_5	Z_6	Z_7	Z_8	Z_9
S_1	.6985	.1523	.0898	.0406	.0141	.0038	.0008	.0001	0
S_2	.3014	.3970	.1523	.0898	.0406	.0141	.0038	.0008	.0001
S_3	.1491	.1523	.3970	.1523	.0898	.0406	.0141	.0038	.0008
S_4	.0593	.0898	.1523	.3970	.1523	.0898	.0406	.0141	.0046
S_5	.0187	.0406	.0898	.1523	.3970	.1523	.0898	.0406	.0187
S_6	.0046	.0141	.0406	.0898	.1523	.3970	.1523	.0898	.0593
S_7	.0008	.0038	.0141	.0406	.0898	.1523	.3970	.1523	.1491
S_8	.0001	.0008	.0038	.0141	.0406	.0898	.1523	.3970	.3014
S_9	0	.0001	.0008	.0038	.0141	.0406	.0898	.1523	.6985

Table 16

Conditional Probabilities ($P(Z_k|S_i)$) for U = .70

State of Nature	Sample Estimate Z_k								
S_i	Z_1	Z_2	Z_3	Z_4	Z_5	Z_6	Z_7	Z_8	Z_9
S_1	.6664	.1387	.0964	.0558	.0269	.0109	.0036	.001	.0003
S_2	.3336	.3328	.1387	.0964	.0558	.0269	.0109	.0036	.0013
S_3	.1949	.1387	.3328	.1387	.0964	.0558	.0269	.0109	.0049
S_4	.0985	.0964	.1387	.3328	.1387	.0964	.0558	.0269	.0158
S_5	.0427	.0558	.0964	.1387	.3328	.1387	.0964	.0558	.0427
S_6	.0158	.0269	.0558	.0964	.1387	.3328	.1387	.0964	.0985
S_7	.0049	.0109	.0269	.0558	.0964	.1387	.3328	.1387	.1949
S_8	.0013	.0036	.0109	.0269	.0558	.0964	.1387	.3328	.3336
S_9	.0003	.001	.0036	.0109	.0269	.0558	.0964	.1387	.6664

Table 17

Conditional Probabilities ($P(Z_k|S_i)$) for U = .60

State of Nature	Sample Estimate Z_k								
S_i	Z_1	Z_2	Z_3	Z_4	Z_5	Z_6	Z_7	Z_8	Z_9
S_1	.633	.1187	.0943	.067	.0423	.0239	.012	.0054	.0032
S_2	.3668	.2662	.1187	.0943	.067	.0423	.0239	.012	.0086
S_3	.2481	.1187	.2662	.1187	.0943	.067	.0423	.0239	.0206
S_4	.1538	.0943	.1187	.2662	.1187	.0943	.067	.0423	.0445
S_5	.0868	.067	.0943	.1187	.2662	.1187	.0943	.067	.0868
S_6	.0445	.0423	.067	.0943	.1187	.2662	.1187	.0943	.1538
S_7	.0206	.0239	.0423	.067	.0943	.1187	.2662	.1187	.2481
S_8	.0086	.012	.0239	.0423	.067	.0943	.1187	.2662	.3668
S_9	.0032	.0054	.012	.0239	.0423	.067	.0943	.1187	.633

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